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LAND USE IN LCA

Is land use impact assessment in LCA applicable for forest biomass value chains? Findings from comparison of use of Scandinavian wood, agro-biomass and peat for energy

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Abstract
Purpose A framework for the inclusion of land use impact assessment and a set of land use impact indicators has been recently proposed for life cycle assessment (LCA) and no case studies are available for forest biomass. The proposed methodology is tested for Scandinavian managed forestry; a comparative case study is made for energy from wood, agro-biomass and peat; and sensitivity to forest management options is analysed.
Methods The functional unit of this comparative case study is 1 GJ of energy in solid fuels. The land use impact assessment framework of the United Nations Environment Programme and the Society of Environmental Toxicology and Chemistry (UNEP-SETAC) is followed and its application for wood biomass is critically analysed. Applied midpoint indicators include ecological footprint and human appropriation of net primary production, global warming potential indicator for biomass (GWP_{bio}-100) and impact indicators proposed by UNEP-SETAC on ecosystem services and biodiversity. Options for forest biomass land inventory modelling are discussed. The system boundary covers only the biomass acquisition phase. Management scenarios are formulated for forest and barley biomass, and a sensitivity analysis focuses on impacts of land transformations for agro-biomass.
Results and discussion Meaningful differences were found in between solid biofuels from distinct land use classes. The impact indicator results were sensitive to land occupation and transformation and differed significantly from inventory results. Current impact assessment method is not sensitive to land management scenarios because the published characterisation factors are still too coarse and indicate differences only between land use types. All indicators on ecosystem services and biodiversity were sensitive to the assumptions related with land transformation. The land occupation (m^2a) approach in inventory was found challenging for Scandinavian wood, due to long rotation periods and variable intensities of harvests. Some suggestions of UNEP-SETAC were challenged for the sake of practicality and relevance for decision support.
Conclusions Land use impact assessment framework for LCA and life cycle impact assessment (LCIA) indicators could be applied in a comparison of solid bioenergy sources. Although forest bioenergy has higher land occupation than agro-bioenergy, LCIA indicator results are of similar magnitude or even lower for forest bioenergy. Previous literature indicates that environmental impacts of land use are significant, but it remains questionable if these are captured with satisfactory reliability with the applied LCA methodology, especially for forest biomass. Short and long time perspectives of land use impacts should be studied in LCA with characterisation factors for all relevant timeframes, not only 500 years, with a forward-looking perspective. Characterisation factors need to be modelled further for different (forest) land management intensities and for peat excavation.

Keywords Barley · Bioenergy · Indicators · Land use · LCIA · Peat · Reed canary grass · Wood

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1 Introduction

With the on-going expansion of human population and continuous increase in material well-being, discussion has arisen on the planetary boundaries on, for example, climate change, loss of biodiversity and use of productive land (Rockström et al. 2009). Aspiration towards bioeconomy and the increased use of renewable energy, including bioenergy, has been considered in the European Union policy as one of the possible solutions to mitigate human impact on the environment, especially climate (Directive 2009/28/EC, EC 2011). However, biomass feedstock provision is much more land use intensive than production of majority of traditional non-renewable feedstocks. With the on-going competition between forestry, agriculture, infrastructure and natural ecosystems over productive land, a discussion has arisen on the consequent environmental impacts from increased land use connected with bioeconomy (e.g. Searchinger et al. 2008; Hertel et al. 2010; Barona et al. 2010).

Life cycle assessment (LCA) has been widely accepted as the methodology for environmental impact assessment of products and services (Finnveden et al. 2009) and the United Nations Environment Programme and the Society of Environmental Toxicology and Chemistry (UNEP-SETAC) Life Cycle Initiative has proposed a framework for the inclusion of land use-related environmental impact assessment in LCA (Milà i Canals et al. 2007a; Koellner et al. 2013a). The UNEP-SETAC work has enabled the development of many midpoint and endpoint land use indicators for life cycle impact assessment (LCIA), for example Schmidt (2008) and de Baan et al. (2013a, b) on biodiversity, Brandão et al. (2010) and Brandão and Milà i Canals (2013) on soil quality and biotic production and Müller-Wenk and Brandão (2010) on climate regulation potential (CRP). Additionally, Ewing et al. (2010) and Haberl et al. (2007) have introduced independent ecological indicators that can be applied in LCA as midpoint indicators on competition over productive land and net primary production (NPP). Although the LCA methodology has been increasingly implemented for biomass value chains in order to evaluate their environmental impacts (see e.g. Cherubini and Strømman 2011; Cespi et al. 2013), only a few peer-reviewed LCA case studies have applied the land use impact assessment framework and indicators (e.g. Michelsen 2008; Mattila et al. 2012; Michelsen et al. 2012; Coelho and Michelsen 2013; Milà i Canals et al. 2013; Núñez et al. 2013). Results in Mattila et al. (2012) indicated a high sensitivity to land transformation flows in the studied beer product system while Milà i Canals et al. (2013) came into a conclusion that land occupation is the main driver for environmental impacts in the studied margarine product system. These partially contradicting conclusions and the limited availability of published LCA case studies highlight the need to test the proposed land use impact assessment framework and land use indicators further in a multitude of LCA case studies for different land uses in different biomes. Moreover, majority of the published LCA case studies have focused on agro-biomass, while no published studies have had a focus on the land use impact assessment of forest biomass value chains with a broad set of land use impact indicators. Michelsen (2008) and Michelsen et al. (2012) do present forest bioenergy value chains but are limited to biodiversity and climate indicators.

Building on this background, the goals of this study are to:

- Test the applicability of the proposed land use impact assessment framework and a selected set of land use LCIA indicators for biomass value chains with a special focus on biomass from managed Scandinavian forests in the boreal region.
- Find out whether the methodology can find meaningful differences in the land use impacts for different sources of energy, including wood, barley, reed canary grass and peat.
- Test the sensitivity of the method to differing forest management options.
- Make suggestions for further research needs, if identified.

2 Methods

This study focuses on the land use impact assessment within the framework of LCA for solid biomass that is readily available in Scandinavia for combustion for heat and power. The main sources of biomass for energy in Scandinavian countries are currently forest biomass and peat (ECON Pöyry 2008), while possibilities for expanding the use of agro-biomass, such as barley grain, straw and reed canary grass for heat and power production, have been studied (e.g. Mäkinen et al. 2006). The functional unit of this comparative case study is 1 GJ of energy in solid biofuels measured as lower heating value (LHV). Potential end-use for the solid biofuel is considered to be combustion for heat in a residential power plant. As the main focus is to test the applicability of the land use LCIA framework (Milà i Canals et al. 2007a; Koellner et al. 2013a) and LCIA indicators for forest land use, a set of management scenarios for forest biomass acquisition is applied. This study follows the attributional approach for life cycle inventory (LCI) modelling.

2.1 Land use impact indicators

To be able to distinguish between environmental relevance of, for example, occupation of 1 ha of mineral excavation site, managed forest land or arable land for annual crops, impact assessment is needed. Different characterisation factors (CFs) have been developed for LCIA for different land use classes and for different environmental mid- and endpoints and areas

of protection (see e.g. review of indicators in Mattila et al. 2012). Following the grouping proposed by Ridoutt et al. (2013) for midpoint land use impact characterisation, the impact indicators are divided into resource-based indicators and ones indicating impacts on ecosystem services and biodiversity. A resource-based approach to land use impact modelling considers that productive land is a scarce and limited resource (cf. Rockström et al. 2009) and that the production of any goods or services adds incrementally to the global demand for productive land, thus having an impact on loss of natural ecosystems (Ridoutt et al. 2013). Examples of such indicators are ecological footprint (Ewing et al. 2010) and human appropriation of net primary production (HANPP) in Haberl et al. (2007). In terms of driver–pressure–state–impact-response (DPSIR) framework (see e.g. Stanners et al. 2007), resource-based indicators would represent pressure on natural ecosystems, not direct impacts. Regarding impact indicators for ecosystem services and biodiversity, a special issue of this journal titled 'Global land use impacts on biodiversity and ecosystem services in LCA' (editors: Koellner and Geyer 2013) has recently proposed a set of land use impact assessment models and includes CFs for biodiversity damage potential (de Baan et al. 2013a) and several ecosystem services (see e.g. Brandão and Milà i Canals 2013; Saad et al. 2013). In terms of the DPSIR framework, this latter subset of indicators can be considered representative of potential environmental impacts on individual ecosystem services and on biodiversity. The pressure and impact of land use LCIA indicators applied in this study are listed below and presented in further detail in Sections 2.1.1–2.1.2.

- Direct ecological footprint = land use footprint (LUF)
- Human appropriation of net primary production (HANPP)
- Biotic production potential (BPP)
- Freshwater regulation potential (FWRP)
- Erosion resistance potential (ERP)
- Water purification potential by physiochemical filtration (WPP-PCF) and by mechanical filtration (WPP-MF)
- Global warming potential via impacts on terrestrial C cycle with 100 year timeframe ($GWP_{(bio)}$-100)
- Biodiversity damage potential (BDP)
- Potentially lost non-endemic species, regional (PLNS)

Please note that the abbreviation CF refers to *characterisation factors* throughout this study and should not be confused with the term carbon footprint. Climate impacts are referred to with abbreviation $GWP_{(bio)}$-100 throughout this study. All the CFs applied in this study are available in Online Resource (ESM).

2.1.1 Resource (pressure) indicators

Ecological footprint indicator quantifies the demand that humans put on natural capital in terms of occupation of biological productive area (Wackernagel et al. 2002) and is measured in hectares normalised to the average productivity of all bioproductive hectares on Earth (Ewing et al. 2010). Ecological footprint has been operationalized previously in LCA for example in Huijbregts et al. (2008) and Mattila et al. (2012). Ecological footprint indicator is defined in Ewing et al. (2010) as the sum of direct land occupation (EF_{direct}) and indirect land occupation through the need for carbon uptake of fossil greenhouse gas emissions (EF_{CO_2}). The EF_{direct} term of any studied system is measured in abstract units, global hectares, and is based on both the actual land area and the equivalence (weighting) factors of the bioproductivity of specific land use types occupied (see Ewing et al. 2010, Table 2). In this study, only EF_{direct} was considered to be relevant as resource indicator for products, following the notion made by Steen-Olsen et al. (2012) that the carbon uptake land can be considered to overlap with carbon footprint indicator. EF_{direct} is termed as land use footprint (LUF) throughout this study. Equivalence factors present in Ewing et al. (2010) for EF_{direct} were applied as CFs in this study with the exception that artificial land was assumed to be most often built on forest land in the boreal region. Thus, CF of forest land (instead of cropland in Ewing et al. 2010) was applied for artificial land use in Scandinavia. It should be stressed that EF_{direct} (LUF) differs from life cycle inventory items such as 'agricultural land occupation' and 'urban land occupation' that have been proposed to be used as midpoint land use indicators without any characterisation (Goedkoop et al. 2008). EF_{direct} (LUF) indicator differentiates the bioproductivity of distinct land use types. For example, the occupation of 1 ha of forest land is considered to put less pressure on the availability of productive land than the occupation of 1 ha of agricultural land.

Human appropriation of net primary production (HANPP) describes the difference in the free NPP left for ecosystems between the current land use and a reference natural state (Haberl et al. 2007). HANPP indicator highlights how much pressure we apply on ecosystems by indicating how large a share of NPP we appropriate for our uses. HANPP serves as a pressure indicator on the use of limited resource (bioproductivity). Additionally, the amount of free NPP left for the ecosystem has been found to correlate well with species diversity. CFs for HANPP in the boreal region are applied in this study based on the method described and CFs published in Mattila et al. (2011, 2012).

2.1.2 Indicators on impacts on ecosystem services and biodiversity

For biotic production potential (BPP), the characterisation model first presented in Milà i Canals et al. (2007b) and later adopted in Brandão and Milà i Canals (2013) was applied.

The CFs for the boreal region present in Brandão and Milà i Canals (2013) were used in this study. The model is known to be very sensitive to the underlying uncertain data on soil organic carbon (SOC) stocks in different land uses and is especially sensitive to the selection of SOC level in the reference state (see discussion in e.g. Müller-Wenk and Brandão 2010; Mattila et al. 2012). Thus, care needs to be applied in the interpretation of the results. For freshwater regulation potential (FWRP), erosion resistance potential (ERP), water purification potential by physiochemical filtration (WPP-PCF) and by mechanical filtration (WPP-MF), the approach and CFs for the boreal region present in Saad et al. (2013) were applied.

Regarding climate impact assessment of biomass acquisition, a modification of the global warming potential approach for bioenergy (GWP_{bio}-100) that was first proposed by Cherubini et al. (2011) and later modified by Pingoud et al. (2012) was applied in this study. The GWP_{bio}-100 factors present in Pingoud et al. (2012) were applied for forest bioenergy as those have been modelled in line with the UNEP-SETAC land use impact characterisation model (Milà i Canals et al. 2007a), that is, considering natural regeneration in no-harvest reference baseline (cf. Helin et al. 2013). The selection of GWP_{bio}-100 as the characterisation model for climate impacts of land use via impacts on terrestrial C cycle is a deviation from the recommendation in Koellner et al. (2013a) to apply the climate regulation potential (CRP) approach and respective CFs proposed by Müller-Wenk and Brandão (2010). It was decided not to follow this suggestion for two reasons: First, the main focus of this study is forest biomass and Müller-Wenk and Brandão (2010) does not fully grasp the dynamics of carbon stock changes in managed forest land use (see Helin et al. 2013 and the discussion section of this paper for more details). The second reason was that the CFs present in Müller-Wenk and Brandão (2010) are modelled for a rather long timeframe (500 years) from the climate mitigation perspective, which limits the relevance of the indicator results for today's decision-making purposes. GWP_{bio} factors that depict impacts via changes in terrestrial C cycle with a timeframe of 100 years were considered to have more policy relevance. For Scandinavian agro-bioenergy, that is, reed canary grass and barley grain and straw in this study, the average of GWP_{bio}-100 factors modelled and present in Koponen and Soimakallio (2013) were applied in this study. The GWP_{bio}-100 factors applied in this study do not consider other greenhouse gases (GHG) than CO_2, although other GHGs might be relevant especially for peat and agro-biomass (Soimakallio et al. 2009). Thus, $GWP_{(bio)}$-100 factors applied in this study depict climate impacts only through changes inferred on terrestrial C cycle. This simplification was adopted in line with the suggested CRP approach (Müller-Wenk and Brandão 2010) and the potential impacts of this limitation are discussed in Section 3. CO_2 originating from peat combustion was given GWP-100 factor of fossil CO_2 in accordance with IPCC (2006), although one could argue that the GWP_{bio} (or GWP_{peat}) approach could be applied for peat biomass as well. IPCC (2006) does not define peat as fossil fuel nor as biomass, but as peat, and states that 'Although peat is not strictly speaking a fossil fuel, its greenhouse gas emission characteristics have been shown in life cycle studies to be comparable to that of fossil fuels [...] Therefore, the CO_2 emissions from combustion of peat are included in the national emissions as for fossil fuels' (IPCC 2006, p. 1–15). Thus, according to IPCC (2006) the GWP-coefficient of fossil CO_2 shall be applied in the emission *accounting* in the energy sector (e.g. retrospective national GHG emission reporting) when peat is combusted for energy. Kirkinen et al. (2008) have implemented an approach similar to GWP_{bio} for peat and their results indicate that the radiative forcing (climate impact) of peat energy use (including all life cycle GHG emissions) is of similar magnitude as of hardcoal combustion in LHV basis in a 100-year timeframe. Thus, the simplified assumption of considering CO_2 from peat combustion equal to fossil CO_2 in life cycle impact assessment perspective is justified.

Impacts of land use on biodiversity were studied with two indicators, biodiversity damage potential (BDP, de Baan et al. 2013a) and potentially lost non-endemic species (PLNS, de Baan et al. 2013b). For BDP, the species diversity approach present in de Baan et al. (2013a) was adopted in this study. As no data points are included in that study for the boreal region, there was a need to apply CFs for some other geographical area. The global level CFs for BDP published in Milà i Canals et al. (2013) were applied in this study, as it is the only source that considers both land occupation and transformation flows. The PLNS indicator aims at describing the potential regional extinction of non-endemic species that are considered reversible. The mean value for aggregated species approach was applied from de Baan et al. (2013b) for the ecoregion Scandinavian and Russian taiga (PA0608). This ecoregion covers the majority of land area in Finland, especially the areas suitable for managed forestry, agriculture and peat excavation.

2.2 LCI system boundary and data sources

This comparative case study on solid biofuels considers the land occupation and transformation flows connected with the acquisition of following biomass fractions suitable for energy in Scandinavian conditions:

- Long rotation forest stemwood
- Peat sourced from peat excavation bogs
- Barley with two management scenarios: harvest of barley grain only or harvest of both barley grain and straw
- Reed canary grass

Straw, barley grains, stemwood biomass and peat have similar properties as combustibles, such as elementary

composition, moisture content and lower heating value (LHV) (Alakangas 2000). Therefore, they can be combusted for energy in the same boiler in heat and/or power plant and can be considered to provide the same function in energetic use on LHV basis. Reed canary grass has different properties as solid biofuel and it can only be considered to be co-fired with other biomass (Alakangas 2000). Despite of slightly different function, reed canary grass was included in this study as it has one of the highest yields of commercially cultivated agro-biomass in Scandinavian climate (Koponen and Soimakallio 2013), thus indicating the theoretical optimum for field bioenergy.

As one aim of this case study is the comparison of land use impacts of solid bioenergy sources, it was considered adequate to focus on the differences in land use in the bioenergy value chains. These value chains include inputs for silviculture, agriculture and peat excavation, area reserved for biomass growth and harvest, biomass transport and end-use as energy in boiler. As the power plant is identical for all the studied energy sources, the land use flows of the use phase were omitted. Moreover, the land use LCA case study for barley-based beer in Mattila et al. (2012) showed that the contribution of agricultural inputs and biomass transports in land use inventory and impacts in barley value chain was shown to be insignificant in comparison with the direct land use in barley cultivation phase. As the case study for vegetable oils in Milà i Canals et al. (2013) indicates similar results, and as there are no major inputs such as fertilisers in silviculture nor peat bogs, both transports and production of agri/silvicultural inputs were omitted from the studied system. Thus, the system boundary of the case study was reduced to cover the land use flows and climate impacts from the biomass acquisition phase only.

2.2.1 Net calorific values of solid biofuels

Net calorific values (lower heating values, LHV) in solid biofuels need to be determined for the studied biofuels. LHV per dry matter in fuel (LHV_{dm}) can be calculated as a function of higher heating value (HHV) and moisture content

$$\mathrm{LVH_{dm}} = \mathrm{HHV} - \left(\frac{m_{\mathrm{H_2O}}}{m_{\mathrm{dm}}} \times L\right), \qquad (1)$$

where m_{H_2O} is the mass of water in the fuel, m_{dm} is the mass of dry matter in the fuel and L the latent heat, 2.443 MJ/kg_{H_2O} , required for evaporation of water in normal pressure and temperature. The HHV of softwood biomass is ca. 20 MJ/kg_{dm} and a typical moisture content is 50 %, and the HHV of milling peat is ca. 21.6 MJ/kg_{dm} and a typical moisture content is 50 % (Alakangas 2000). When this data is applied in Eq. (1), LHV_{dm} values of 17.6 and 19.2 MJ/kg_{dm} are determined for wood and peat, respectively. Mäkinen et al. (2006) include LHV data for reed canary grass, barley grain and straw. LHV_{dm} of 17.6 MJ/kg_{dm} (in 20 % moisture) was applied for reed canary grass and LHV_{dm} of 17.1 MJ/kg_{dm} for barley grain and straw (Mäkinen et al. 2006). These are summarised in Table 1.

2.2.2 Land occupation and transformation flows

The land occupation flow in biomass acquisition is a function of regional biomass yields. The determination of annual harvested biomass yield is not as straightforward for wood biomass as it is for field biomass, as the rotation period for forest biomass in Scandinavia is decades or even a century, and as the annual stemwood growth is not constant over the rotation period of one stand. One can consider the total biomass growth of one stand over rotation period and calculate a theoretical mean annual increment, or one can derive actual annual yields from statistical data on regional level on managed forest land area and actual annual harvests taking place. The former approach for determining stemwood yield assumes that the annual increment in the forests equals annual harvests in the region (and is applied in e.g. ecoinvent 2.0, see Warner et al. 2007), while the latter approach respects the fact that the annual increment and harvests can differ significantly depending on the actual intensity of forest biomass harvests (i.e. the standing stock in forests can follow an increasing or a declining trend). As there has been a continuous trend of increasing standing stocks in European forests, including Scandinavia (Köhl et al. 2011), the latter approach for determining regional average forest biomass yield was applied in this study. It was considered to be more representative of the actual situation, instead of the theoretical maximum sustainable wood yield in the competing approach. Actual growth and harvest data for Southern Finland in the year 2010 was applied based on the Statistical Yearbook of Forestry (Finnish Forest Research Institute 2011). There is approximately 11 million ha of productive managed forest land available for forestry in total in Southern Finland, while the annual total harvests in 2010 were approximately 45.7 million m^3 (Finnish Forest Research Institute 2011, Chapter 1, Table 1.1 and Chapter 4, Table 4.16), resulting in average yield of ca. 4.2 m^3/ha*a. Data in Köhl et al. (2011) enables the determination of actual yields in any region in Europe, if necessary in future studies.

Additionally, to study the impact of possibly changing the intensity of harvests, mean annual yield data was applied for four future forest management scenarios for 2010–2110[1]: silviculture as usual (SAU), active forest sector and intensive

[1] Data on future forest management scenarios have been produced in Finnish Forest Cluster project EffFibre and were provided as written notification from Jari Hynynen from the Finnish Forest Research Institute METLA.

Table 1 Life cycle inventory parameters for all the studied bioenergy scenarios

Biomass category and scenario	Yield [m^3/ha*a]	Land occupation [m^2a/kg_{dm}]	LHV [GJ/kg_{dm}]	L.occ per f.u. [m^2a/GJ]	L.transf. to/f.u. [m^2/GJ]	CO_2 intensity [$kgCO_2$/GJ]	$GWP_{(bio)}$-100 factor [unitless]
Softwood (2010)	4.2	6.1	17.6	348	–	110[a]	0.6[b]
Softwood (INT)	6.4	4.0	17.6	228	–	110[a]	0.6[b]
Softwood (QLTY)	6.3	4.1	17.6	234	–	110[a]	0.6[b]
Softwood (SAU)	5.3	4.8	17.6	274	–	110[a]	0.6[b]
Softwood (EXT)	3.5	7.2	17.6	410	–	110[a]	0.6[b]
Peat	–	0.16	19.2	8	0.5	106[a]	1
Barley grain	–	3.28	17.1	192	–	110[a]	0.9[c]
Barley grain and straw	–	2.43	17.1	142	–	110[a]	0.9[c]
RCG, no LUC	–	1.67	17.6	95	–	100[a]	0.5[c]
RCG, LUC from forest	–	1.67	17.6	95	4.7	100[a]	–
RCG, LUC from urban	–	1.67	17.6	95	4.7	100[a]	0.5[c]

RCG reed canary grass, *LUC* land transformation

[a] Statistics Finland (2011)

[b] Pingoud et al. (2012)

[c] Koponen and Soimakallio (2013)

biomass production (INT), decreasing activities of forest management and increasing non-material services (EXT) and forest management that aims towards high quality raw material production for forest industry and bioenergy (QLTY). The average annual wood removals for the scenarios are, in decreasing order, 6.4 m^3/ha*a for INT scenario, 6.3 m^3/ha*a for QLTY scenario, 5.3 m^3/ha*a for SAU scenario and 3.5 m^3/ha*a for EXT scenario. It was assumed that softwood is used for bioenergy as the majority of harvested and standing forest biomass in Scandinavia is softwood. When the density of Norway spruce, 390 $kg_{drymatter}/m^3_{wood}$ (Jyske et al. 2008) is applied, the land occupation flows for stemwood biomass are (all dry matter basis):

- 6.1 m^2a/kg_{wood} for current (2010) state
- 4.0 m^2a/kg_{wood} for INT scenario
- 4.1 m^2a/kg_{wood} for QLTY scenario
- 4.8 m^2a/kg_{wood} for SAU scenario
- 7.2 m^2a/kg_{wood} for EXT scenario

The difference in the scenarios highlights that the intensity of harvests has a strong influence on land occupation flows of forest biomass acquisition and the related challenges are discussed in more detail in Section 3 of this study. No land transformation flows were identified for wood biomass acquisition from sustainably managed Scandinavian forests.

Majority of peat biomass is typically acquired by the milled peat method from forestry-drained peatlands in Scandinavia (Kirkinen et al. 2008). The question of determining the annual yield of slow growth biomass unambiguously is present for both wood and peat biomass. Natural mires accumulate peat biomass with a rate that can deviate significantly from the average annual peat biomass harvests in peat excavation bogs. The same approach is applied for peat as was applied above for wood biomass. Approximately 63,000 ha of mires are under peat excavation activities in Finland each year, and the net area has remained relatively constant in recent years (Alakangas 2000). One excavation bog can be used for approximately 15–20 years; thus, it can be assumed that ca. 5–7 % of annual peat excavation area is abandoned every year and the same amount is newly converted to maintain the constant production rate. In recent years, ca. 20–29 TWh of peat has been extracted annually, depending mainly on the annual climatic conditions (too wet conditions in a mire prevent mechanical excavation activities). An average value of 25 TWh peat harvested annually was applied in this study in the calculation of average annual peat yield. This results in ca. 7–8 m^2a/GJ peat LHV basis in fuel (or 0.14–0.16 $m^2a/kg_{dry\ mass}$ in peat) and 8 m^2a/GJ is applied as a conservative estimate. One challenge in determining the land occupation flows of peat energy is the fact that land use classification does not have any class that fully represents the nature of the activity. Land use class 'occupation, mineral excavation site' was applied, as it is the one with most identical aspects. Consequent challenges regarding the land use impact characterisation are discussed in Section 3. It was assumed that new excavation sites are transformed from forestry-drained peatlands (transformation from forest, extensive) and the same area of peat excavation sites is abandoned and afforested for managed forestry (transformation to forest, intensive), based on the peat biomass value chains presented in Kirkinen et al. (2007). A conservative estimate, 7 % of the occupied land area

transformed annually, was applied in this study and it equals ca. 0.5 m^2/GJ land transformed. It needs to be noted that the *net* natural land transformations due to peat excavation activity are 0 (net land transformation from 'forest, extensive' to 'forest, intensive').

Barley can be harvested once a year in the summer season in Scandinavian climate and the cropland has low productivity during the remaining of the year (often covered with snow). Thus, the arable land area occupied for barley cultivation can be completely allocated to annual crop of barley. The average yields in Mäkinen et al. (2006) for barley grains and straw, 3,045 and 1,066 kg_{dm}/ha*a, respectively, were applied in this study. Total annual yield in the barley grain and straw for bioenergy scenario is thus 4,111 kg_{dm}/ha*a. This results in arable land occupation flows of 3.28 and 2.43 m^2a/kg_{dm} for barley grains only and barley grains+straw scenarios, respectively. Reed canary grass is a perennial crop and an average annual yield of ca. 6,000 kg_{dm}/ha (Mäkinen et al. 2006) was applied for reed canary grass in this study. The respective land occupation flow is of 1.67 m^2a/kg_{dm} for reed canary grass.

Agro-bioenergy can be considered to result in direct or indirect land transformation flows (e.g. Kløverpris et al. 2008), with a large degree of uncertainty in the estimates (Plevin et al. 2010). In the base case agro-biomass scenarios, it was assumed that no land transformations take place. However, in order to test the sensitivity of land use impacts of agro-biomass production to the assumptions on land transformations, a sensitivity analysis was applied for best and worst case scenarios for reed canary grass acquisition. The worst case scenario includes 100 % direct land transformation from unmanaged forest and the best case scenario 100 % direct land transformation from artificial (sealed) urban land use, e.g. a parking lot transformed to agriculture. Initial land transformation is amortised to first 20 years, following the suggestion in Koellner et al. (2013a). In other words, land transformation flow allocated to annual harvests is 5 % of direct land occupation flow, 0.083 m^2/kg_{dm} reed canary grass. All life cycle inventory parameters are summarised in Table 1.

2.2.3 Greenhouse gas flows and GWP_{bio}-100 factors

The only LCI flow required in the climate impact assessment of bioenergy with the GWP_{bio}-100 approach that depicts impacts via changes in terrestrial C cycle is the direct CO_2 emission from combustion of biomass in the power plant. Land occupation inventory, inverse indicator of biomass yield, is not required as it is already considered in the modelling of GWP_{100}-bio factors (Koponen and Soimakallio 2013). Climate impacts of forest biomass combustion are directly connected with the rotation length of trees and carbon content removed from the forest and only indirectly connected with the land area required for wood acquisition. In other words, if growth parameters (climatic conditions, habitat, etc.) are kept constant, climate impacts are a function of the total biomass harvested, irrespective of whether it was sourced from a small area in intensive harvests or from a large area in extensive harvests. As it is known that the intensity of annual harvests varies, thus forest land occupation per unit of harvested forest biomass varies, less uncertainty is present when the CFs are connected with amount of biomass, not land area occupied. Direct CO_2 emission factors and GWP_{bio}-100 factors for all studied solid biofuels are present in Table 1 (Statistics Finland 2011; Pingoud et al. 2012; Koponen and Soimakallio 2013). Koponen and Soimakallio (2013) include only an uncertainty range (high and low end estimates) for GWP_{bio}-100 factors of agro-bioenergy. The average value of each agro-biomass species was applied in this study.

No GWP_{bio}-100 factors were found in the literature for agro-bioenergy in the case of land transformation taking place in reed canary grass production system. Calculation of new CFs for land transformation flows for agro-bioenergy is out of the scope of this study; thus, none were applied. This would be relevant only for the case of forest-to-cropland transformation in sensitivity analysis for reed canary grass, as the carbon stock in artificial and cropland can be considered equal (Müller-Wenk and Brandão 2010), thus having no impact on carbon stocks. GWP_{bio}-100 factors for land transformations can be calculated in future studies by appreciating the equations in Koellner et al. (2013a, Fig. 1) and the concept of forward-looking perspective with flexible selection of impact modelling timeframe presented in Section 3.3 of this study.

3 Results and discussion

3.1 Impact assessment of solid biofuels

The results of impact assessment for studied solid biofuels are presented in Table 2, and graphic illustrations of the results of each individual indicator can be found in Online Resource (ESM Figs. S1–S8). Both the differences in the results between distinct energy sources and the differences in the results between management scenarios of specific biomass are of interest, and these two viewpoints are discussed separately.

3.1.1 Different sources of energy

Meaningful differences could be found in between distinct solid biofuels that were originating from different land use classes (arable, managed forest or mineral excavation area). Forest biomass occupies (in all management scenarios) larger land areas per functional unit than agro-biomass, and peat occupies the smallest area. The LCIA indicator results did differ significantly from these inventory results as all the applied indicators have different CFs (differentiated impacts)

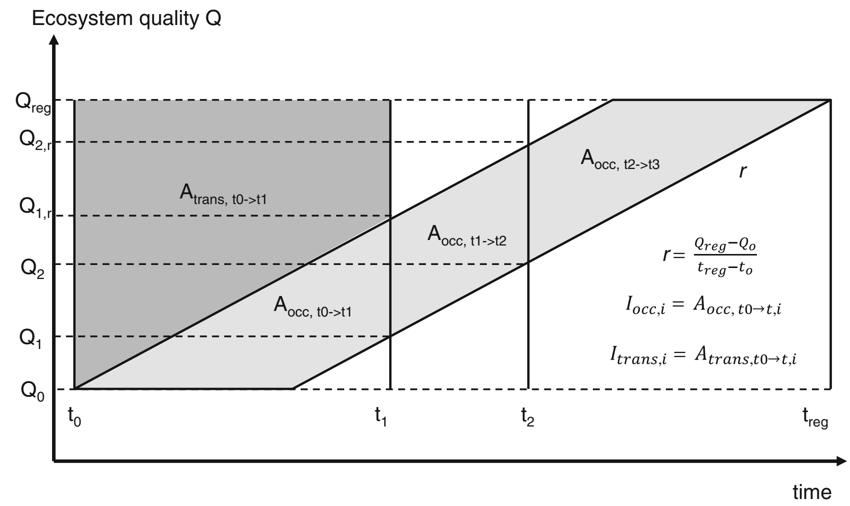

Fig. 1 Illustration of modelling of land occupation (*light grey area*) and transformation (*dark grey area*) impacts in forward-looking perspective with flexible selection of impact modelling timeframe. *I* environmental impact, *A* area, *occ* occupation, *trans* transformation, t_i time *i*, *0* initial state, *reg* full regeneration to quasi-natural state, *r* regeneration rate. If the applied timeframe is from t_o to t_{reg} (or beyond), only then $I_{occ}=\Delta Q=Q_{reg}-Q_0$

for these three land types. Thus, the case study reveals that no conclusions nor comparisons should be made on environmental impacts of land use based only on the land use inventory results (m^2a and m^2 converted per functional unit). Land use impact assessment is needed.

Resource-based indicators LUF and HANPP give consistent results with each other (Table 2, Figs. S2–S3), although they measure pressure on productive land resource and consequently pressure on natural ecosystems with slightly differing characterisation models. Both give more weight to agricultural and mineral excavation land occupation than for forest land occupation, which leads to smaller differences between forest and agro-biomass than what the inventory result would suggest. Peat seems to have the lowest pressure on natural ecosystems when measured with resource-based indicators and reed canary grass the second lowest. No significant difference could be found with resource (pressure) indicators in between forest and barley biomass, and the relative difference between the two is mainly dependent on the management scenario.

BPP indicates soil quality and potential future biomass productivity. BPP indicates that agro-bioenergy has larger impacts on future biomass production potential than what peat energy does (Table 2, Fig. S4a). Additionally, there is no impact from forest biomass use. The result can be questioned regarding peat, as the CF for mineral excavation had to be used for peat biomass excavation. Peat actually *is* SOM by definition, and its excavation activity should have a differentiated CF for the BPP indicator. On the other hand, in peat excavation sites, the depth of SOM layer is from a few meters to tens of meters, and the productivity is largely dependent on the SOM content in the top soil and depth of the water layer (oxygen intake of roots) in the mire. Calculation of new CFs is out of the scope of this study, but is suggested for future studies. The BPP indicator highlights the impact of possible land use changes, as the results are sensitive to the assumptions on land transformations (Fig. S3).

The ERP indicator is closely related with SOM deficit phenomenon present in the BPP indicator. The ERP indicator results are very similar with the results of the BPP indicator (Table 2, Fig. S4b) as both indicate that agro-bioenergy has the highest negative impact, peat a smaller negative impact and forest bioenergy no or even positive impacts. Additionally, ERP is sensitive to land transformations, although the worst and best case analysis for reed canary grass shows that ERP is more sensitive to the applied transformation flows than BPP. The similarities in results could suggest that ERP and BPP indicators largely overlap and only one could be applied in future studies, at least in the boreal region.

Both ERP and SOM deficit (BPP) indicators correlate with future NPP potential in managed forests and provide meaningful results. However, these are only two of a set of suitable indicators for future NPP potential in managed forests, together with nutrient balance, soil pH and soil compaction (Hansen et al. 2011). Harvesting of biomass in managed forest land is known to potentially lead to loss of nutrients, with more pronounced impacts if the share and amount of extracted biomass is increased (intensification of harvests). An example is residues removal in whole tree harvesting. This has been found to have an adverse impact on the future productivity of the forest (Helmisaari et al. 2011). Thus, it seems that ERP and SOM deficit are suitable indicators for ecosystem service of future NPP, but do not capture the most influential impact pathway regarding future NPP/BPP in Scandinavian managed forests: nutrient balances. New CFs should be developed to capture actual differences caused to future NPP by changed harvesting intensity.

Regarding WPP, the internally normalised results (Table 2, Fig. S6) reveal that both WPP-MF and WPP-PCF indicators yield identical results with each other at least in the boreal region. This could indicate that it is sufficient to consider

Table 2 Land use impact assessment results for all studied bioenergy scenarios. Positive indicator values signify detrimental impacts and vice versa

Biomass category and scenario	L. occ. (LCI) [m^{2}a]	LUF [gm^{2}a]	HANPP [kgC]	BPP [kgC*a]	ERP [kg soil]	FWRP [mm g water/a]	WPP-MF [cm/day]	WPP-PCF [cmol$_c$/kg soil]	GWP-100 [kgCO$_{2,foss}$eq]	BDP [% species]	PLNS [pot. lost species*1e-9]
Indicator values[a]											
Softwood (2010)	348	438	52	0	−45	2,333	588	382	67	63	1.28
Softwood (INT)	228	287	34	0	−29	1,530	385	251	67	41	0.84
Softwood (QLTY)	234	294	35	0	−30	1,568	395	257	67	42	0.86
Softwood (SAU)	274	345	41	0	−35	1,836	462	301	67	49	1.01
Softwood (EXT)	410	517	61	0	−53	2,754	694	451	67	74	1.52
Peat	8	10	3	686	65	1,057	259	169	106	5	4.50
Barley grain	192	482	57	5,070	500	2,212	325	211	99	115	1.33
Barley grain and straw	142	357	43	3,755	370	1,639	240	156	99	85	0.99
RCG, no LUC	95	238	28	2,500	246	1,091	160	104	50	57	0.66
RCG, LUC from forest	(a)	(a)	(a)	3,750	776	2,648	160	104	(c)	186	79.8
RCG, LUC from urban	(a)	(a)	(a)	−13,620	−4,250	−301,080	−124,280	−80,950	50	116	65.8
Internally normalised[b]					(b)						
Softwood (2010)	100 %	100 %	100 %	0	−100 %	100 %	100 %	100 %	100 %	100 %	100 %
Softwood (INT)	66 %	66 %	65 %	0	−66 %	66 %	66 %	66 %	100 %	65 %	66 %
Softwood (QLTY)	67 %	67 %	67 %	0	−67 %	67 %	67 %	67 %	100 %	67 %	67 %
Softwood (SAU)	79 %	79 %	79 %	0	−79 %	79 %	79 %	79 %	100 %	78 %	79 %
Softwood (EXT)	118 %	118 %	118 %	0	−118 %	118 %	118 %	118 %	100 %	117 %	118 %
Peat	2 %	2 %	6 %	(c)	144 %	45 %	44 %	44 %	158 %	8 %	350 %
Barley grain	55 %	110 %	110 %	(c)	1,111 %	95 %	55 %	55 %	148 %	183 %	104 %
Barley grain and straw	41 %	82 %	83 %	(c)	822 %	70 %	41 %	41 %	148 %	135 %	77 %
RCG, no LUC	27 %	54 %	54 %	(c)	547 %	47 %	27 %	27 %	75 %	90 %	51 %
RCG, LUC from forest	(a)	(a)	(a)	(c)	9,466 %	113 %	27 %	27 %	(c)	295 %	6,213 %
RCG, LUC from urban	(a)	(a)	(a)	(c)		−1,730 %		−12,095 %	−21,150 %		−21,164 %
75 %	184 %	5,126 %									

(a) Indicator considers land occupation LCI flows only, (b) negative value = positive impact, (c) not computable

L.occ land occupation LCI, *LUF* land use footprint, *HANPP* human appropriation of net primary production, *BPP* biotic production potential, *ERP* erosion resistance potential, *FWRP* freshwater regulation potential, *WPP* water purification potential by mechanical filtration (*MF*) or by physiochemical filtration (*PCF*), *GWP-100* global warming potential via impacts on terrestrial C cycle in 100 years, *BDP* biodiversity damage potential, *PLNS* potentially lost non-endemic species

[a] Indicator results per gigajoule (LHV) in solid biofuel

[b] Indicator results are internally normalised to softwood (2010)

impacts on WPP with only one indicator, especially if this is the case in all biomes. As the CF for managed forest and agricultural land is identical, the indicators suggest that forest bioenergy has higher negative impact on potential water purification than agro-bioenergy. The impact of peat is of similar magnitude with agro-biomass, because CF for artificial land, including mineral excavation sites, is considerably higher than for forest and arable land. The CF for urban cover or mineral excavation site might not describe peat biomass excavation activity in peatbogs properly; thus, care should be taken in the interpretation of the results. In the context of land use and the water purification ecosystem service, a question arises on what detrimental compounds are actually present in different land uses that need to be purified (e.g. in agriculture and managed forests). Both WPP indicators do correctly indicate similar WPP per land area; thus, the highest impact on WPP ecosystem services occurs in forest bioenergy system, but it remains questionable if this indicator result is relevant for decision-making; that is, is there a difference in the presence and input of unwanted compounds to be purified in forest and cropland soil, and consequently, a difference in the water quality of seepage water in the downstream water receptors? This WPP indicator result is contradicting with the prevailing understanding that forest streams and groundwater are of good quality in comparison with water from other land uses (Gundersen et al. 2011) and that agricultural land is afforested in Northern Europe as part of a strategy to restore protective functions, thus improve water quality and secure water resources for the future (Hansen et al. 2007).

The FWRP indicator leads to similar results with WPP indicators in the comparison of studied bioenergy sources (Table 2, Figs. S5–S6), with the exception that barley has impacts of similar magnitude as forest biomass with respect to the majority of the management scenarios. It is questionable whether the FWRP indicator highlights meaningful impacts on ecosystem services, especially regarding forest land in moist boreal region. One of the major ecosystem services of forests is the provision of water in sufficient quality and quantity. The reduced groundwater recharge rate (millimeter groundwater/time) is connected with the large water storage capacity of forest ecosystems and provides an ecosystem service of regulating freshwater flow rates like a sponge does, that is, decreasing peak flow during rainy periods and increasing base flow during dry periods (Katzensteiner et al. 2011). Although FWRP correctly indicates that managed forests reduce the groundwater recharge rate, it is highly controversial if a higher FWRP indicator score signifies an increased or decreased impact on forest ecosystem services, especially in humid boreal region where water flow regulation is a major ecosystem service provided by (managed) forests and where the impact of forest management on water yield may be negligible (Katzensteiner et al. 2011).

The two indicators proposed for biodiversity by de Baan et al. (2013a, b) led to a notable discrepancy in the results for the studied energy carriers (Table 2, Fig. S8). The BDP indicator suggests that peat fuel has the smallest direct impact on biodiversity through land use and agro-bioenergy the largest, while the PLNS indicator would suggest completely the opposite. There seems to be three potential drives for this deviation. First is that the PLNS indicator is much more sensitive to land transformations than the BDP indicator. In case that there are no land transformations connected with biofuels, reed canary grass has a similar (with BDP) or lower (with PLNS) impact on biodiversity in comparison with forest biofuels. With transformations considered, the impact indicator results for reed canary grass increase 2- to 3-fold with the BDP indicator while they increase 100- to 120-fold with the PLNS indicator. The significantly different result for peat fuel is a consequence of the same driver where the PLNS indicator gives significantly more weight to the biodiversity impact from net land transformation from ‘forest, extensive’ to ‘forest, intensive’. The second potential driver for the discrepancy in the results is the relative difference in CFs between forest and agricultural land occupation flows. Both indicators consider that occupation of 1 ha of agricultural land has a higher impact on the number of species than occupation of equal area of (intensively) managed forest does. But larger relative difference in the impacts in between these two land use types is present in the BDP indicator, thus indicating higher relative impacts for agricultural land occupation. Additionally, it needs to be noted that no regionally differentiated CFs were present in de Baan et al. (2013a) for BDP for the boreal region, while for PLNS, the ecoregion-specific CFs were available and applied. Thus, the representativeness of the BDP indicator results is uncertain for biomass from Scandinavia due to lack of CFs for the boreal region. Consequently, in-depth comparison of the results is not meaningful at this point. Additionally, a challenge for the interpretation of the results is that a relative unit [percent] is used in the BDP indicator. The standard practice of summing the impacts in LCIA phase leads to BDP results of deficit of species in percentages that exceed 100 %, which is counter-factual, thus problematic for interpretation. If, for example, the functional unit of this study was 10 GJ instead of 1 GJ, almost all the BDP indicator results would result in hundreds or thousands of percentages. Thus, the absolute unit approach (number of species potentially lost) applied in the PLNS indicator seems to be preferable over the relative unit approach applied in BDP. The use of relative CFs would call for weighted average approach in LCIA modelling in order to yield intuitive indicator results (deficit of 0 to 100 % in species richness), but this is not often supported by commercial LCA software tools.

Measured with BDP, peat seems to have the lowest pressure on natural ecosystems and biodiversity when land use is considered. However, it should be noted that the peat

excavation (or the initial drainage of the peatland for forestry purposes some decades ago in Finland) changes the mire ecosystem to a forest. Even though the area needed is relatively small, the loss of the habitat and mire species is permanent. In that sense, PLNS seems to give more meaningful result for peat energy use. Also, when the impacts of forest management are considered, it would be important not only to estimate the amount of species present, but the change in quality of the habitats as well. The challenge is that forestry in boreal areas changes the habitats and the species composition of forests very slowly. These indicators do not take into account the time lag in species response to habitat quality loss (e.g. the extinction debt), which means the future extinction of species due to past or present activities (Hanski and Ovaskainen 2002). Species richness as an indicator has been criticised for not taking into account all the aspects of biodiversity (see details in de Baan et al. 2013a). The amount of species is not enough to describe the changes in the amounts of individuals of the species and the rarity of the species. Rare species may be replaced by generalist ones, which cannot be assessed with this indicator. As stated by de Baan et al. (2013a), at this point of the development of the BDP indicator, the result should be considered only ‘as first screening of potential land use impacts across global value chains’. These indicators are a good starting point for assessing impacts of different energy sources on biodiversity in global land use impact assessments to identify the optional alternatives among land use types. In order to assess impacts on biodiversity more specifically, other dimensions of biodiversity, such as ecosystem quality, should be considered. For instance, that could be done by evaluating the conditions maintained for biodiversity as suggested by Michelsen (2008).

All the studied bioenergy sources, not only peat, were shown to have warming climate impacts when the GWP_{bio}-100 factors that depict impacts via changes in terrestrial C cycle were applied (Table 2, Fig. S7). Peat has the most significant climate impact, 106 $kgCO_{2,fossil}eq/GJ_{fuel,dm}$; barley biomass very similar, 99 $kgCO_{2,fossil}eq/GJ_{fuel,dm}$; stemwood biomass slightly lower impact, 67 $kgCO_{2,fossil}eq/GJ_{fuel,dm}$; and reed canary grass the lowest warming impact, 50 $kgCO_{2,fossil}eq/GJ_{fuel,dm}$. Only average estimates of GWP_{bio}-100 factors of significant underlying uncertainty (cf. Koponen and Soimakallio 2013) were applied in this study; thus, the large underlying uncertainty is not highlighted in the point estimate approach for LCIA adopted in this study. Additionally, no impacts from land transformations (land use change) were considered in this study, and the climate impact assessment focused on land carbon stocks and CO_2 only. Results in Soimakallio et al. (2009) would suggest that N_2O emissions in agro-biomass cultivation are the largest single source of uncertainty in the estimates of climate impacts of agro-bioenergy. Thus, special care needs to be applied in the interpretation of the results of this case study. The reason that bioenergy was found to have warming climate impacts is that GWP_{bio}-100 factors applied in this study depict climate impacts of land (biomass) use relative to a non-use natural regeneration reference (cf. Milà i Canals et al. 2007a; Müller-Wenk and Brandão 2010). Traditionally, this phenomenon has not been considered for bioenergy sources in attributional LCA case studies (see e.g. review in Cherubini and Strømman 2011), although LCA methodology aims at describing relative potential environmental impacts of activities, and the ILCD handbook (JRC-IES 2010, Chapter 7.4.4.1) and Milà i Canals et al. (2007a) specifically call for inventory modelling in relation to a ‘no-use’ of the studied site. GWP_{bio}-100 factors applied for agro-biomass (Koponen and Soimakallio 2013) include some relevant modifications to the approach and CFs presented in Müller-Wenk and Brandão (2010), especially regarding modelling of radiative forcing (actual warming impact) and modelling of regional natural relaxation rates, and consider the most commonly applied 100-year timeframe in climate impact assessment, instead of 500 years. Additionally, Müller-Wenk and Brandão (2010) do not differentiate carbon stocks of natural and managed forests, out of which the latter are known to have smaller carbon stocks (e.g. Luyssaert et al. 2008). Thus, GWP_{bio}-100 factors that respect the increase in forest carbon stocks in a ‘non-use’ baseline (Pingoud et al. 2012) were preferred over the CRP indicator. If this phenomenon and impact assessment approach is respected, then neither agro- nor forest bioenergy can be considered completely climate neutral. For comparison, direct warming impacts (GWP-100) at power plant, with no life cycle considerations included, are 55 $kgCO_{2,fossil}eq/GJ_{fuel}$ and 108 $kgCO_{2,fossil}eq/GJ_{fuel,dm}$ for natural gas and brown coal, respectively.

The significant difference in the inventory and LCIA indicator results confirms that the LCIA results are not solely driven by land occupation LCI, as was the case for margarine in Milà i Canals et al. (2013). Our results confirm the findings of Mattila et al. (2012) that land occupation inventory result alone is not a sufficient proxy for environmental impacts when different land use types are involved and that the applied impact assessment indicators are sensitive to land transformation flows. It proved to be necessary to include land use impact indicators in LCA in order to understand the potential differences in environmental impacts of land use of different sources of bioenergy.

3.1.2 Sensitivity to management options

The applied methodology found significant differences in the environmental impacts in different management scenarios for specific bioenergy sources, for example for forest biomass (Table 2). However, it needs to be questioned whether these differences are meaningful and actually represent the reality. Land use impact assessment results for forest biomass would suggest that the more intensive the harvests, the less environmental impacts are caused through land use. CFs are available

for forest in level 3 'forest, intensive' (cf. Koellner et al. 2013b), but scenarios reveal a need for further division under intensively managed forests. Currently available impact assessment method is not sensitive to land management scenarios due to the fact that the published CFs are still on a coarse level in an early phase of their development and can only find differences between different land use typologies. The most intensively managed system always yields in the lowest environmental impact scores, irrespective of the fact that intensifying management most probably leads to increased environmental impacts per hectare. This is clearly highlighted in the counter-factual results that a change in barley harvesting from grain only to grain and straw would lead to a decrease in HANPP, ERP and SOM (BPP) deficit per functional unit, although the extraction of straw from the fields most probably leads to increasing or neutral impacts in these pathways. Milà i Canals et al. (2013) came into the same conclusion on too coarse level of published CFs for example for annual and perennial oil crops. It is evident that this issue is not dependent on the product system studied, but the deficiency of the method will prevail until CFs are modelled and published for different management options in future studies. No differentiation between reed canary grass (perennial) and barley (annual) was possible in this study as the published CFs for most impact categories do not differentiate these two types of arable land. Additionally, level 3 classification (cf. Koellner et al. 2013b) is not adequate for managed forests and further division is needed.

All the studied impact indicators on ecosystem services and biodiversity consider transformation flows, while resource (pressure) indicators LUF and HANPP capture only land occupation flows (Table 2). Impact indicators on ecosystem services and biodiversity were found to be very sensitive to the assumptions related with land transformation flows in the reed canary grass biofuel system. Most of the indicators show similar, high sensitivity to the best and worst case transformation scenarios, with the exception of the BDP indicator. BDP is also sensitive to the existence of land use change phenomenon, but the results differ from ecosystem service indicators for artificial-to-cropland transformation. Ecosystem service impact indicators show very positive (beneficial and orders of magnitude higher) impacts in case of artificial-to-cropland transformation, while BDP leads to an increased negative (detrimental) impact on biodiversity in comparison with the base reed canary grass case. The high sensitivity found in this case study for the assumptions on land transformation flows is very different from the findings and conclusions made by Milà i Canals et al. (2013) in land use LCA case study for oil crops. This suggests that the relative impact of land occupation and transformation flows is very dependent on the case studied and that land occupation flows are not as dominating as the results of Milà i Canals et al. (2013) would suggest. This is in line with the findings made in Mattila et al. (2012) land use LCA case study on beer production.

3.2 Land use impact assessment framework and forest biomass value chains

Many challenges were faced in the land use impact assessment of bioenergy from managed forests. The intensity of annual wood harvests varies, thus forest land occupation per unit of harvested forest biomass accordingly, but the impacts on ecosystem services and biodiversity most probably do not remain constant per hectare with different intensities of wood harvests. Consequently, the generally accepted occupation approach (m^2a) in inventory modelling suggested by Milà i Canals (2007a) and Koellner et al. (2013a) seems to be problematic at the moment for forest biomass, because only static impact CFs for 'forest, managed' are currently available for the impact assessment phase. There are two possibilities to potentially resolve this issue in future studies: If forest inventory modelling approach that respects the different intensities of wood harvests (adopted in this study) is followed, then differentiated CFs should be modelled and published for differing intensities of forest land use and management (level 4 in the classification in Koellner et al. 2013b) in future studies. More certainty might be reached in a second option, in which impact assessment would be connected to inventory flow of forest biomass harvested (cubic meter or kilogram) instead of managed forest land area occupied (square meter per year). Many impacts are probably more closely connected to the amount of forest biomass harvested than the forest land area that the forest biomass was sourced from. Alvarenga et al. (2013) have proposed such an approach in the context of land use impact assessment of productive land as a resource in LCA, in which they propose a division of LCI flows into human-made (m^2a) and natural systems (cubic meter) in inventory modelling. The division into two is of course not unambiguous as nominally 'wild' natural systems are almost non-existent and largely influenced by human activities (cf. Hobbs et al. 2006). Managed forests are the most evident example of land use that does not clearly fit either one category, and land use impacts of wood from Scandinavian managed forests are most probably different from wood-harvesting activities in pristine forests. The second option for forest land inventory modelling considers a theoretical mean annual wood biomass increment (square meter per year) as the LCI flow, towards which the impacts on ecosystem services are modelled in the LCIA phase. This theoretical yield (m^2a) equals to a large degree with unit of wood harvested (cubic meter). The approach that respects actual intensity of wood removals was considered more representative of the actual forest land occupation in this study, but the latter approach could help to avoid counter-factual results, that is,

artefacts of static impact modelling methods (generalised CFs) in future studies on forest biomass. Although some of the land use impacts, such as climate impacts, might be a direct function of the amount of wood harvested, it remains unclear for the authors of this study whether this is the case for all the ecosystem service and biodiversity impact pathways. These two possibilities should be further explored in future studies on wood biomass value chains.

Previous literature on impacts of forest management on ecosystem services and biodiversity (see e.g. Raulund-Rasmussen et al. 2011) suggests that forest management has an impact on forest biodiversity (De Jong et al. 2011) and the forest carbon stocks are lower than in unmanaged forests (Luyssaert et al. 2008). Biodiversity and carbon stocks are on lower levels in managed forests than in natural state, but higher than in other land use types of humans. Regarding soil quality, forest management can have an impact on future BPP through the impact pathway of changes in forest nutrient balances in case of intensive whole tree harvesting management (Helmisaari et al. 2011) and SOM deficits are of less concern. Additionally, (managed) forests provide the ecosystem service of provision of water in sufficient quality and quantity (Katzensteiner et al. 2011) and help in the prevention of erosion (Hansen et al. 2011). An obvious disparity can be observed between previous literature on impacts of forest management and the results obtained in this study on impacts on ecosystem services and biodiversity with the tested LCA methodology and the set of LCIA indicators (or CFs available currently only on a generalised level). Further modelling work on relevant impacts and CFs is needed for managed forest land use.

A large uncertainty is present in the results if the CFs lately published for background systems are applied in LCIA. Koellner et al. (2013a) suggest that the published CFs for impacts on ecosystem services should be applied to background systems and case- and site-specific CFs could be modelled to foreground system with the method described in Saad et al. (2013). This suggestion poses practical challenges, as carrying out LCA studies is a resource-consuming practice already in the present form, and transferring the responsibility of modelling of case-specific CFs to the LCA practitioner does not help with this regard, thus discouraging broader application (cf. Baitz et al. 2013). Moreover, biomass production is usually part of the background system in many biomass value chains in the globalised markets, and from the perspective of the commissioner of an LCA study, foreground system is typically the final processing facility of the biomass (e.g. brewery, margarine manufacturing site or pulp and paper mill) with low or non-existent land use impacts. Thus, the suggestion of Koellner et al. (2013a) can be challenged, and further research and publication work on spatially differentiated CFs up to levels 3 and 4 of land use and management classification (Koellner et al. 2013b) is suggested for the LCIA method developers in the academia.

3.3 Time considerations in reference state setting and impact modelling

Koellner et al. (2013a) explicitly propose that a biome-dependent (quasi-)natural land cover is used as a reference in land use impact characterisation. At the same time, it seems to be implicitly considered that it is a static situation that *was* present in the region or can potentially be reached only *in the distant future*, as it is stated in Koellner et al. (2013a) that no consideration of modelling period is necessary for occupation impacts. Stemming from this, it is further stated that policymakers interested of protection of current environmental quality might not be interested in knowing how far we are from an idealistic, quasi-natural situation and might find the impact indicator results irrelevant for decision-making[2] (cf. discussion in Milà i Canals et al. 2013). Milà i Canals et al. (2013) find it problematic that the proposed set of indicators gives more weight to land occupation that has large distance to the potential (ideological) natural vegetation, instead of land transformation interventions that cause abrupt negative impacts in e.g. terrestrial C stocks and biodiversity today. This problem is evident if the impact modelling period for land occupation impacts is fixed as 500 years for all the impact categories, as suggested by Koellner et al. (2013a) and Müller-Wenk and Brandão (2010) for climate regulation potential, but it does not necessarily mean that the framework could not be used with more relevant timeframes to support actual decisions.

To overcome this issue, one can consider a forward-looking perspective in the determination of the reference situation, select an impact modelling timeframe that is relevant for today's decision-making (e.g. 20, 50 or 100 years for climate impacts) and still respect the quasi-natural land cover as the reference situation (see Fig. 1). In other words, to be able to give relevant support for decision-making, the LCA practitioner could select the most relevant timeframe for impact characterisation in the goal definition stage and apply the CFs modelled for both land occupation and transformation with the respective timeframe in the study (e.g. GWP-100 for climate impact assessment of land use). This can be carried out in full accordance with the proposed guidelines (Milà i Canals et al. 2007a; Koellner et al. 2013a) if the reference situation is considered to be natural relaxation (regeneration) onwards from the current state, not the (quasi-)natural situation that was present in the history or can only be reached within centuries (Fig. 1). Such approach is described for climate impact assessment of managed forest land use systems in Helin et al. (2013) and for agro-bioenergy in Koponen and Soimakallio (2013) and can be applied for any land use type or

[2] LCA is a tool to support decision-making. A decision-maker may be e.g. a policy maker, corporate manager or a consumer, and the relevant decision-making context should be clear for the LCA practitioner in defining the goal and scope of the study.

impact category. *If* the most relevant impact modelling timeframe for the decision support in question is longer than the regeneration time, *then* the CFs applied describe absolute distance to an idealistic natural vegetation state (ΔQ in Eq. (4) in Koellner et al. 2013a). This might be the case for some decision-making support situations for many of the land use impact pathways, but most probably is not relevant for decisions related e.g. with climate regulation and mitigation. The equations for characterisation of transformation and occupation impacts (Koellner et al. 2013a, Eqs. (2) and (4)) could be modified to be a function of regeneration rate and modelling timeframe (r and $t_{0->i}$ in Fig. 1) instead of regeneration time t_{reg}.

Koellner et al. (2013a) stress that the selection of modelling timeframe is a value choice which can have an impact on the relative weight of studied transformation and occupation activities. As the results and discussion in Milà i Canals et al. (2013) suggest, a choice of a 500-year impact modelling period gives more relative weight to land occupation interventions, while a shorter timeframe would most probably increase the relative weight of land transformation activities in the impact indicator results (cf. Fig. 1). As the time lag between land use intervention and its environmental impact can be large, Milà i Canals et al. (2007a) suggest that impacts from land use should be calculated over a reasonable time period, thus supporting selection of a long timeframe for modelling. At the same time one needs to consider, that for an ecological indicator to be effective, it should provide relevant information about changes, be sensitive, be able to detect changes at the appropriate temporal and spatial scale, be based on well-understood and generally accepted conceptual models of the system, be based on reliable data that are available to assess trends and are collected in a relatively straightforward process, be based on data for which monitoring systems are in place and be easily understood by policymakers (Millennium Ecosystem Assessment 2005). There seems to be a trade-off between securing that all the far-reaching impacts are covered (long-term perspective) in land use LCIA and maintaining sensitivity to short-term changes, thus of relevance to today's decision-making purposes. Both short and long time perspectives could be studied in LCA context by applying CFs for all relevant timeframes in any case study (e.g. GWP-20, GWP-100 and GWP-500).

4 Conclusions and recommendations

A consistent framework for land use impact assessment in LCA exists and several LCIA methods and characterisation factors (CFs) are available that could technically be applied in a comparative case study on forest and other biomass acquisition. A significant difference in the inventory and LCIA indicator results confirms that the land occupation inventory result alone is an inadequate proxy for environmental impacts and that land use impact assessment indicators are necessary. Although forest bioenergy has higher land occupation needs than agro-bioenergy, the land use pressure and impact indicator results are of similar magnitude or even lower for forest bioenergy. As no deforestation is connected with Scandinavian forestry, impacts from land transformation were found relevant mainly for agro-biomass in a sensitivity analysis. Energy use of peat resulted in the lowest impacts in the majority of impact categories due to the low land occupation needs, and a trade-off between high climate impacts and low result in many other land use impact categories was observed. This is most probably the case for all fuels that are excavated below ground, for peat and especially for all fossil fuels.

However, caution needs to be applied in the interpretation of results as the geographical detail in the published CFs is still on coarse level and the static CFs can only identify coarse differences between land *use* types but not meaningful differences in between land *management* options within one land use class. Results would suggest that intensive forestry with higher yields and levels of harvest would have lower pressure on productive land availability and lower impacts on ecosystem services. The existing CFs are still on a too coarse level to identify impacts of changed management, especially with ecosystem service and biodiversity impact indicators. Additionally, care should be taken in the interpretation of the results of this study regarding peat biomass as CFs for land use class 'occupation, mineral excavation site' had to be applied for peat biomass excavation, and this raises many questions regarding the validity of the results.

Although previous literature indicates that the actual environmental impacts through land use are known to be significant, especially for agricultural and artificial 'sealed' use of land, it still remains questionable whether these are captured with satisfactory reliability with the applied LCA method and LCIA indicators, especially for managed forest land use. This was the case especially for indicators on biodiversity and ecosystem services of water purification and regulation. Additionally, an evident overlap was observed between many of the resource and ecosystem service impact indicators.

Recommendations and further research needs:

- Short and long time perspectives of land use impacts should be studied in LCA by applying characterisation factors for all relevant timeframes with a forward-looking perspective for reference situation.
- Modelling of CFs for different management options (1) for all the impact pathways to be able to capture differences in (forest) land management types and intensities, (2) for shorter timeframes than 500 years, especially

GWP_{bio}-factors for transformation flows for agro-bioenergy, and (3) for occupation flow ‘peat excavation’ for all the impact pathways.

- Explore possibilities to include additional and supporting characterisation models on impacts of forest management on ecosystem services and biodiversity.

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